Brushfires in California Wildlands: Ecology and Resource Management
Edited by J.E. Keeley and T. Scott. 1995. International Association of Wildland Fire, Fairfield, WA

Post-Fire Emergency Seeding and Conservation In Southern California Shrublands

Todd Keeler-Wolf

Natural Diversity Data Base, Natural Heritage Division, California Department of Fish and Game,1416 Ninth St., Sacramento, CA 95814
Tel. (916) 324-6857; *Fax* (916) *324-0475; E-mail: tkwolf@gishost.dfg.ca.gov*

Abstract. The post-fire reseeding of wildlands for erosion control has been a topic of debate for a number of years in California. Existing evidence argues against seeding with non-native grasses because of the negative influence on natural biodiversity, its ineffectiveness as an erosion control, and exacerbation of erosion due to community type-conversion. As a result of opposition by public land management agencies, conservation agencies, and private land owners following the southern California firestorms in Fall 1993, there was a substantial reduction in the projected use of seed. However, this may be viewed as temporary until the agencies responsible for post-fire management agree to an integrated approach incorporating the most recent scientific findings. After an analysis of the effects of seeding on the natural system in southern California shrublands, I suggest a more focused approach to post-fire management. This focused approach aims at reducing reliance upon seeding as the default treatment. Instead it would implement more strategic erosion control methods based on an assessment of the wildland fire area considering hydrology, geology, soils, vegetation, fire intensity, fire timing, and sensitive natural resources in the light of reduction of losses of life and property. In this new context, post-fire seeding would be limited to specific situations and would involve only native species known to be appropriate to the site.

Keywords: Genetic contamination; Italian ryegrass; native species; postfire erosion; type-conversion; seeding.

Introduction

The goal here is to provide a broad picture of what the environmental and ecological effects are – from genetics to ecosystem functioning – of the artificial seeding practices commonly implemented following a wildland burn. Thus, much of this paper is a synthesis of portions of several papers presented in this volume on post fire ecosystem management. Using this information as background I will conclude with a re-assessment of the use and value of emergency re-seeding following fires.

At present there is much concern from resource management agencies, conservation organizations and other environmental groups that the use of large quantities of non-native grass seeds to establish a cover following fire over bare erodible soil is an ecologically unsound and relatively ineffective mechanism to curb erosion.

Following several major fire events over the past decade conservation organizations such as The California Native Plant Society, The National Park Service, the California Department of Parks and Recreation, and the California Department of Fish and Game have expressed concern over the widespread implementation of non-native seeding, not only in southern California scrublands, but in other ecosystems as well (Ordano 1987). However, until the southern California fires of Fall 1993, post-fire seeding has continued to be used extensively as an erosion control method by the agencies charged with the responsibility of protecting property and lives.

Due to the objections of agencies and organizations with conservation agendas, there was a substantial reduction in the originally projected use of seeding as rehabilitation immediately following the southern California firestorms of October-November 1993. Yet, despite the fact that agencies such as the California Department of Forestry, the Soil Conservation Service, and certain local fire departments are investing much time and money into experiments and studies designed to make better judgements about seeding and its values, they and some other responsible agencies are not convinced that the evidence yet exists to make such judgements (D. Neff, D. Wickheiser personal communications 1994).

The logic of relying upon plants as a relatively low cost and low-technology means of preventing loss of property and soil seems eminently reasonable. The controversy over artificial seeding following a fire has arisen concomitantly with environmental awareness over the past few decades. In particular, the concern for using non-native grasses in high density over large areas has given rise to the current debate.

Background on Post-fire Seeding

When artificial seeding following fires began in southern California in the 1920's and 30's resource managers relied on native species of shrubs such as Ceanothus *(Ceanothus* spp.), chamise *(Adenostema fasciculatum),* toyon *(Heteromeles arbutifolia)* and California buckeye *(Aesculus californica)* (Barro and Conard 1987). These shrub seeds were collected in areas adjacent to the fires and hand sown on them after the fires. The shrubs established well, but did not establish as quickly and extensively as the existing native resprouting species or the native annual herbs.

In an attempt to establish the quickest, most predictable cover of soil-binding vegetation, numerous species were experimented with in the 1920's, 30's, and 40's. Annual grasses were selected as the most useful species because they built the fastest networks of fibrous, stabilizing roots in the upper levels of the soil (Barro and Conard 1987).

Annual Italian ryegrass *(Lolium multiflorum)* was selected as the premier seeding grass in the 1940's (Barro and Conard 1987) for its rapid germination and its ready availability. It also is typically short-persisting at a site, and thus, was thought to not strongly affect the long term balance of the shrub ecosystem (Gautier 1982, Papanstasis 1973, Papanstasis and Biswell 1975). It had been cultivated for many years as a pasture grass and several strains were available (Edmusson and Cornelius 1961, Nelson 1980). Its use proliferated in the 1950's and 60's and by the 1970's very large areas in southern California as well as in more northerly montane areas of the State were being seeded after fires with this species (Anonymous 1970).

Research on the natural ecological dynamics of California shrublands also began in the 1940's and picked up in the following two decades (e.g., Sampson 1944, Horton and Kraebel 1955, Sweeney 1956, Wells 1962, Patric and Hanes 1964). However, this work began well after the perceived need to take a jump on nature was established. Unfortunately, the dissimilarities between the natural regeneration mechanisms and the situation imposed by post-fire grass seeding was not immediately apparent. The ramifications of the poor match between the imposed and natural cycle have only recently been receiving the attention they deserve.

Background on the Natural Fire Cycle of California Shrub lands

The fire cycle in chaparral and coastal sage scrub is distinctive and characterized by an herbaceous phase in the first wet season after fire. Many of the native herbs are adapted specifically to the fire cycle and can survive high intensity fires with their germination being enhanced by various fire effects such as heat and charcoal (e.g., Keeley 1991, 1994).

These herbs are present in great profusion and diversity in the first year or two after a fire (Hanes 1977, Keeley and Keeley 1984, 1988, Malanson and O'Leary 1982, O'Leary 1995). The annuals, like the non-native annual grasses, send out rapid root growth and tend to hold the topsoil better than the seedlings of the shrubs that will eventually come to dominate (Rundel and Parsons 1983, S. Conard personal communication 1994). Although the herbaceous fire-following component of the scrub fire cycle now comprises some non-native (mostly European) species, including some annual grasses such as Zorro fescue *(Vulpia myuros)* and red brome *(Bromus madritensis* ssp. rubens), originally there were few native annual grasses represented in the post fire annual flor. These were principally two species of annual fescues, *Vulpia microstachys* and *V. octoflora,* which although locally common on some bums, probably did not ever dominate (Aikens and Lonard 1993, Sweeney 1956).

Both coastal sage scrub and chaparral have shrub species that resprout and shrub species that only reproduce by seed after fire. There is variation in sprouting ability within many shrub species depending on fire intensity, frequency, and seasonal timing (Zedler 1995, Borchert 1995, and Keeley 1995). However, there are also basic ecological differences between these two plant formations (Westman 1982, O'Leary 1990, Keeley and Keeley 1988). Coastal sage scrub is made up predominantly of soft-leaved (malacophyllous), partially drought-deciduous shrubs, whereas chaparral shrubs are leathery-leaved (sclerophyllous). Coastal sage scrub is more drought-tolerant and can grow on poorer soils than chaparral. Unlike most chaparral shrubs, many dominant species in coastal sage scrub apparently can continue to replace themselves through seeding many years after a fire. This imposes more structural diversity (including vertical and horizontal components to structure) to coastal sage scrub than in typically even-aged chaparral. Coastal sage scrub is usually dominated by smaller stature shrubs than chaparral, but is composed of species with higher levels of volatile oils than chaparral. Thus, coastal sage scrub produces lower fuel levels, but is more flammable at an earlier stage of succession than chaparral (Zedler et al 1983, O'Leary 1995, Keeley 1995).

The vegetation of both coastal sage scrub and chaparral is highly varied depending on slope exposure, local climate, soil type, and other environmental variables. The most recent ecological classifications of these vegetation types divide them into scores of plant associations, each of which have distinct environmental correlations. In the south coastal part of California the new California Native Plant Society classification (Sawyer 1994) lists 13 series of coastal sage scrub with 27 associations and 34 series of chaparral with 82 associations. Some practical implications of this diversity are that two adjacent slopes may have a significantly different suite of plant species (and thus animal species) associated with them, and that there may be significant variation in vegetation from a north-facing slope at, for example, 150 m elevation along the coast than a site with the same exposure and elevation several km inland.

Impact of Annual Grass on the Natural Ecosystem

The competitive balance may already be tipped toward non-native annuals without seeding.

There is some evidence in southern California scrub habitats supporting the invasiveness of alien grasses even without intentional introduction of non-native species of grasses. In Riverside County, Edith Allen (personal communication 1994) has observed the replacement of native coastal sage scrub shrubs by red brome and other non-native European grasses. Many of the European annual grasses are nitrophiles and may be taking advantage of nitrogen pollution (from car emissions primarily) in southern California, occupying interstitial spaces between native shrubs and herbs and out-competing native seedlings. Although its effect on annual grasses is unknown, air pollution has a negative effect on many of the dominant species of coastal sage scrub (O'Leary 1990). Other persistent alien non-grass species, virtually ubiquitous in the wildlands of southwestern California today (such as the mustards *Brassica nigra*, and *Hirschfeldia incana)* may have been widely established with the aid of post-fire seeding in the first half of this century (J. Beyers, personal communication 1994).

Competitive and inhibitory effects of Italian ryegrass

When ryegrass is established after a fire there are demonstratable detrimental effects on the establishment of native species of herbs and the seed-reproducing shrubs. Schultz and Biswell (1952), Schultz et al. (1955), Keeley et al. (1981), Odion and Nadkarni (1985), Taskey et al. (1989), Gautier (1982), and Beyers et al. (1993) have shown that growth and survival of native species is reduced by seeded Italian ryegrass. Although competition for moisture has been assumed to be the principal cause for reducing density and vigor of native species in ryegrass-seeded areas (Schultz et al. 1955), Odion and Nadkarni (1985) have shown ryegrass to suppress and kill native annuals and shrub seedlings by out-competing them for nitrogen. Odion and Nadkarni (1985) also have shown that 75% of the nitrogen in the plants in a seeded burn was held in ryegrass. Ryegrass has also received the reputation of being allelopathic (Taskey et al. 1989) and there is some evidence to support this (Naqvi 1969).

Where native annuals dominate after a fire they release nitrogen to the soil early, when woody plants are still active. However, ryegrass decomposes slowly over the dry season and thus, most of its nitrogen is released to the soil when other plants are dormant. The long-term effect is not known, but can be expected to be adverse (Odion and Nadkarni 1985). In addition, if a good cover of ryegrass is established, the competitive effects it has on important native perennial nitrogen-fixing species such as deerweed *(Lotus scoparius)* and various obligate seeding *Ceanothus* species will reduce long-term soil enrichment to the affected stand.

The problem of type-conversion

Following a chaparral or coastal sage scrub fire non-native grasses and annual herbs vary in persistence, but generally will not be eliminated from the system. Even annual ryegrass, touted for its short persistence, can under certain conditions, persist for longer than four years (Beyers et al. 1993). Under conditions of frequent fires or grazing annual, non-native-dominated vegetation has been shown to increase and replace southern California scrublands (Conrad 1979, Beyers et al. 1994, Barro and Conard 1987, Haidinger and Keeley 1993). This "type-conversion" has been documented in many areas of the state (Keeley 1990). If an area is successfully seeded with grasses and then re-burned, many of the obligate seeder shrubs such as *Ceanothus* species will be killed (Zedler et al1983, Gautier, 1982, Odion and Nadkarni 1985). Keeley et al. (1981) showed that fire annuals are also quickly eliminated under such situations because they cannot compete with a dense cover of exotics, since they are dependent on the return of the shrub cover to shade out exotics before another fire.

In contrast to the herbaceous annual and perennial native fire followers, which generally hold too much moisture during the dry season to support fires, grasses dry very quickly and are flash fuel (Zedler et al. 1983). Fires in this type of situation can quickly convert coastal sage scrub or chaparral to annual grassland (Keeley 1990). This relationship has led Gautier and Zedler (1982) to contend that the only reason to seed with ryegrass is for the express intention of type conversion. In addition to development, much of the recent loss of southern California coastal sage scrub is very likely to have come from increased fire frequencies (Tom White, personal communication 1993). Thus, once a non-native grass cover is established, the likelihood of it persisting through increased fire frequency and competitive elimination of native shrubs and herbs is high.

Effectiveness of Erosion Control by Grass Seeding

Short-term vs. long-term effectiveness of ryegrass for erosion control

Krammes and Hill (1963) and Corbett and Green (1965) have shown possible erosion reductions of up to 16% due to ryegrass plantings. Although these studies may demonstrate short-term (first year) reduction in sediment yield they do not indicate long-term erosion control. This is largely because of the short persistence of ryegrass in the ecosystem and because of its inhibitory effects on other long-persisting native species (Barro and Conard 1987, Beyers et al. 1994, Gautier 1982). Conversely, several studies have shown no effect on erosion and even increases in the erosion rates of areas that have been converted from shrubland to grassland.

The principal reason for the lack of any noticeable effect on erosion revolves around the unpredictability of ryegrass germination coupled with the timing of peak erosion in the first year or two immediately after a fire. Several studies indicate poor initial germination due to unfavorable climatic conditions (Corbett and Green 1965, Blankenbaker and Ryan 1985, Beyers et al. 1993). Ryegrass requires relatively high moisture availability for germination and growth (Odion and Nadkarni 1985). The variable Mediterranean climate of southern California produces highly unpredictable rainfall (Clark 1993) and many years does not provide the moisture necessary for successful ryegrass growth and germination. Besides the unpredictability, several studies (Krammes and Hall 1963, Blankenbaker and Ryan 1985, and see Barro and Conard 1987) have shown that in the first year after a fire most of the rainfall occurs before a significant amount of seeded grass has become established and if rain or wind is too intense they may remove substantial amounts of seed from the slopes. Several researchers (e.g., Rice 1974, Wells 1984) have substantiated that in southern coastal California most erosion occurs in the first 1-3 years following a fire. A combination of these points creates serious doubt about the reliability and cost-effectiveness of ryegrass seeding.

Combustion of organic compounds within the burning vegetation creates a vapor which condenses at depth within the soil, forming a water-repellant or hydrophobic soil (Booker et al. 1993). In areas where the soils are strongly hydrophobic the repellant nature of the sub-soil will increase rilling and sheeting erosion, and may inhibit root growth and establishment of the grasses, while at the same time washing much of the seed downslope (p. Wohlgemuth and S. Conard, personal communication 1994). It is the rilling and sheeting erosion that ryegrass seeding has been used to ameliorate (Ruby 1989). However, erosion due to rilling even in strongly hydrophobic soil may not be as significant as originally assumed due to natural fissures and animal holes conducting much water beneath the hydrophobic layer (Booker et al. 1993). Barro and Conard (1987) showed that over a huge area of ryegrass seeding done in 1970 (a year when 70% of the 241,086 ha burned in the State was seeded) cover was extremely variable and native groundcover consistently provided higher cover than ryegrass.

In areas where slopes are steep (>35°) the likelihood of ryegrass establishment by the standard aerial or hand spreader application methods is very low (P. Wohlgemuth personal communication 1994). In such situations native cover has been shown to be higher than ryegrass cover (Blankenbaker and Ryan 1985).

The paradox: increased erosion in ryegrass-seeded areas.

The principal reason for observed reductions in erosion control revolve around the long-term effects of ryegrass seeding on the ecosystem. The processes of dry ravel and associated stream scouring and outwash debris torrents are the principal forms of erosion and soil loss following shrubland fires in southwestern California (Anderson et al. 1959, Rice 1974, DeBano et al. 1979, Wakimoto 1979, Wells 1981, Spittler 1995), accounting for from 50 to 70% of the total erosion in many areas. This type of erosion is also the principal cause of the damaging mudflows so prevalent during storms in fire areas (commonly termed the "fireflood sequence" in the literature, see Booker et al. 1993). Storms of sufficient intensity to move channel sediments occur every 8-10 years in the absence of fire (Rice 1974). Following fire, even normal storms can cause significant erosion. Sediment yields are higher in burned watersheds because of increased runoff resulting from lack of vegetation and an increase in soil hydrophobicity (Gautier and Zedler 1982). Thus, the value of seeding depends mainly on whether it can reduce peak stream flow, and there is no evidence that this is the case (Matthews 1987).

Shrub foliage intercepts rainfall and shrub root systems hold wet soil more effectively than those of grasses (Booker et al. 1993, Barrows et al. 1993, Corbett and Crouse 1968). Hence, if herb and shrub seedlings are out-competed and replaced by grasses under type-conversion conditions, there will be higher rates of dry ravel and more sediment build-up on the same slope with the shrub cover removed.

Loss of shrub seedlings may be especially important since they contribute greatly to rapid recovery of the shrub canopy. Conrad (1979) and Zedler et al. (1983) have demonstrated the long-term negative effects on seeding shrubs following successful ryegrass establishment. For example, Conrad (1979) noted a 25% reduction in shrub cover on watersheds seeded with ryegrass 18 years before compared to non-seeded watersheds. Thus, slowing the recovery of shrub cover will almost assuredly result in increased erosion rates once ryegrass disappears from the vegetation (Gautier 1982, Gautier and Zedler 1982). Although erosion rates later in the fire cycle are relatively low, even a small increase in a fire interval of normally 20-50 years could easily counterbalance any decrease in erosion due to successful grass seeding in the first 1-3 years (Gautier and Zedler 1982).

Under optimum conditions for ryegrass germination, and in areas without strongly hydrophobic soils, grasses will channel more water into the substrate and less will be allowed to runoff than under a shrub-dominated cover (Wright, 1977, Booker et al 1993, Spittler 1993, Barrows et al. 1993). Although this reduces rilling and sheet erosion initially, as we have seen, these are not the principal causes of property damage and soil loss in the fire-flood sequence documented for southern California. In fact, the effectiveness of soil-water infusion by grass cover may frequently be detrimental to the stability of many slopes in southern California.

Soils derived from marine sediments such as shales and sandstones in much of southern coastal California are relatively fine-grained and/or clay rich (Sharp 1978, Barrows et al. 1993). Thus, they are expansive when they become wet. When water infiltrates down the grass blades and stems, these soils quickly become saturated and have a tendency to develop shallow slope failures resulting in massive instability and soil and property loss. The slope failures occur below the reach of the fibrous root systems of grasses (Booker et al. 1993). This is a particularly severe problem in type converted areas where the slope stabilizing properties of deep-rooted shrubs are absent (Corbett and Rice 1966, Barrows et al. 1993,). Converting chaparral to grassland through the burn-seed-reburn technique resulted in a seven-fold increase in erosion on the San Dimas Experimental Forest in the San Gabriel Mountains (Rice and Foggin 1971).

Another reason for the surprising increase of erosion rates in areas where a grass layer is established involves pocket gopher *(Thomomys* spp.) activity. Gophers are well adapted to grass-dominated vegetation and are likely to increase their populations in areas that have been converted from shrub cover to grass cover. The churning of soils via gopher excavations (bioturbation) creates substantial soil loss (Booker et al. 1993). Gophers also follow the spread of grasses into other areas and exacerbate erosion rates there (Taskey et al. 1989). Although bioturbation loss may not be significant compared to erosion due to dry ravel and stream scouring, under certain conditions it has been shown to be responsible for up to 10 times the erosion rates from sheet and rill erosion (Booker et al. 1993).

The long term effects of a successfully established ryegrass cover thus include:

1. a reduction in native biodiversity,
2. in the case of coastal sage scrub, degradation of an already threatened natural community (NCCP 1993), and
3. a higher erosion rate for the site over the long-term, than would have occurred under natural conditions.

Post-Fire Seeding with "Native" Species

Over the past several years ryegrass has been replaced, at least in part, by seed mixes with other species of grasses and forbs that have been touted as being native or "nearly native". The use of non-natives flies in the face of habitat conservation in an era of protection of the few remaining natural areas in the south coastal portion of the State. Conversely, seeding with species native to California or adventive species that are already firmly established within the southwestern California shrub ecosystems might seem to be a reasonable alternative since these could be assumed to occur naturally in the fire zone and thus be a better ecological and environmental fit than clearly introduced species.

Natives versus "new natives"

The question of native authenticity has arisen for several species now commonly included in post-fire seed mixes including Zorro fescue *(Vulpia myuros),* Blando brome *(Bromus hordeaceus),* and Hykon rose clover *(Trifolium hirtum).* Based on distribution records from several regional floras *(Jepson* 1925, Munz 1959, Munz 1974, and Hickman 1993) both *T. hirtum* and *B. hordeaceus* are consistently listed as of European origin. The case for *Vulpia myuros* is less clear. The earlier floras such as Jepson and Munz suggest that although *V. myuros* is a European, the closely related and now conspecific *(fide* Hickman 1993) *V. megalura* is considered a native Californian. However, in the most recent treatment of the genus *Vulpia,* Aiken and Lonard (1983) conclude that both species are likely to have originated in Europe. Anonymous (1993) suggests that Zorro fescue has been in California since the Mission Period.

Despite their non-native status, species such as Zorro fescue have been described by the Soil Conservation Service (Anonymous 1993) as beneficial and excellent in seed mixes because they allow perennials to become established after readily colonizing disturbed areas, thus minimizing long term ecological change. However, visual evidence from the Oakland fire of 1991 shows that after 3 years dense stands of *Vulpia myuros* still exist in certain places where the native Cucamounga (California) brome *(Bromus carinatus)* and California blue wild rye *(Elymus glaucus)* were also seeded (personal observation, April 1994).

Several non-native species that have been present in the California flora for years such as mustards *(Brassica nigra* and *Hirschfeldia incana)* and the star thistles *(Centaurea solstitialis* and C. *melitensis)* are likely to have proliferated as a result of seeding activities over the past several decades, either from intentional seeding or as inadvertent impurities in seed mixes of Italian ryegrass and other species (B. and C. Wilson written communication 1992, and J. Beyers personal communication 1994, Conard and Beyers 1995).

The main point here is that although there are hundreds of non-native species firmly established in the state, and many of them so entrenched that they have been considered as "new natives" (Heady 1977), they are not natural components of the fife cycle in California shrublands. Many of the points discussed above for Italian ryegrass are applicable to any annual grass seeded in a scrub community. The tendency of these shrub communities to be type-converted to weedy annual-dominated plant communities by shortened fire intervals (Gautier and Zedler 1982) is reason enough to avoid intentionally tipping the balance toward non-native dominance. However, chaparral and coastal sage scrub communities also act as valuable watershed cover. The long term value of chaparral versus annual grassland in erosion control has been demonstrated (Corbett and Rice 1966, Corbett and Crouse 1968).

Chaparral and coastal sage scrub contain many rare elements of natural diversity. Over 60 species of vascular plants within the coastal sage scrub vegetation alone are considered by the California Natural Diversity Data Base as being special (either State or Federally listed as rare, threatened, or endangered, or listed by the California Native Plant Society as a list 1b species) status (California Department of Fish and Game, Natural Heritage Division unpublished data 1993). The effect of competition by non-native annuals on many of these species could be disastrous, with or without the consequences of type-conversion.

Use of California natives not known from the burn site

Grading into the issue of native versus non-native seeding is the concept of using species that are known to be native to the state, but are not native to the bum site. The number of scrub associations in southern California (Sawyer 1994) indicates the tendency for vegetation assemblages to change under subtle changes in environmental conditions. Thus, native species not known from a specific site should not be used there. However, many Soil Conservation Service determinations of seed mixes applied on fire sites are made without prior knowledge of the species composition of the site (D. Weirman and D. Dyer personal communication 1994).

The introduction of the "wrong" mix of native species for a burned site could have obvious negative effects including:

1. replacing previous species that were dominant or important members of the pre-fire flora by species that never occurred there,

2. out-competing local endemic species, and

3. altering the long-term ecosystem balance by changing preferred food, breeding or cover species for locally adapted animal life.

The question of genetic appropriateness

As a further extension of the above concept it can be argued that local genetic adaptation across the geographic range of a particular species may be developed sufficiently to warrant careful selection of local ecotypes for certain areas. Thus, a locally adapted population of a species could be genetically diluted or swamped-out by a massive introduction of another genetic strain of the same species, originating from a different area. This genetic appropriateness concern has been raised recently for many vegetation rehabilitation and restoration projects including post-fire seeding (Libby and Rodrigues 1992) and rare plant restoration (USDA Forest Service 1994). After the fall 1993 fires in southern California, frustration was expressed by some fire rehabilitation specialists from the California Department of Forestry and the Soil Conservation Service. This was due to the certainty that a particular dominant species existed at a fire site, that a large seed supply of this species' was available, and yet that land managers or owners elected not to use the seed for fear of genetic contamination.

Although limited studies and genetic theory have indicated that in some cases (e.g., coniferous trees) there is cause for concern of genetic contamination, there has to date been no work on the genetic variation and genetic swamping susceptibility of any major chaparral or coastal sage scrub species. However, it is likely that there is a significant amount of ecotypic variation in chaparral and coastal sage scrub plants (A. Montalvo personal communication 1994). Although the distribution and reproductive biology of species can give us some clues about likely tolerance of introductions of non-local strains, there are exceptions to almost every supposition. If we are to seed with species not collected at or near the site, we need research to understand the consequences of this action. In the interim period it would be prudent to follow a conservative seeding policy for revegetation and restoration similar to that drafted by the U.S. Forest Service for the use of native plant material in restoration and other revegetation projects in California (USDA Forest Service 1994). This policy should assure the use of native seeds only where the origin is known, and that these seeds should be locally adapted, and of high quality.

Rare Species Concerns and Seeding

The likely effects of seeding on bona fide rare species (state and federally listed or candidates for listing) are variable. These depend on, among other things, the habitat, the reproductive biology, and the life form of the rare species in question. For example the several rare *Dudleya* species known from south coastal California typically grow on rock outcrops or steep bluff faces away from high intensity fire situations. Many of these species are also capable of resprouting if they are burned. Thus, their vulnerability to fire would be considered low as would their vulnerability to competitive effects of seeding. Other species such as the annual composite (Asteraceae) *Pentachaeta lyonii,* which is known from fewer than 20 occurrences in openings in chaparral, coastal scrub, and grassland in and near the Santa Monica Mountains may be more vulnerable (Skinner and Pavlik 1994). Species such as this one could loose populations if repeated fires occurred at the wrong times of the year. Such fires could deplete seed reserves while concomitant seeding could foster annual non-natives that would out-compete the less aggressive native annual herbs.

There were 24 taxa of rare plants tracked by the California Natural Diversity Data Base that were affected by the fall 1993 southern California fires. The response of these species to fires and seeding is in general poorly known and probably highly varied. Without detailed population monitoring it is impossible to address the threats to these species as a result of fire and seeding with non-natives. To date, no detailed population studies have been conducted on any of these species (California Department of Fish and Game Natural Heritage Division Endangered Plant Program species management database 1994).

For animals, certainly the gross effects of type-conversion and the somewhat lesser effects of inappropriate seeding on local natural communities could easily reduce the populations of many rare species such as the orange-throated whiptail and the California gnatcatcher. These effects are particularly significant in the already highly fragmented and urbanized mosaic of scrub habitats in southwestern California.

Effects of Hydroseeding Applications

Although aerial seeding from helicopters has been the most widely used method for applying grass seed to fire areas in recent years, because of its inaccuracy and relative ineffectiveness, there are other techniques that are gaining favor. The most widely used alternative seeding technique is hydroseeding. This involves mixing a solution of water, seeds, and a mulch composed of various materials (most commonly, cellulose fibers with a polymer "tackfier" derived from paper pulp) and spraying this solution via high pressure hoses onto slopes. The advantages to this method include:

1. a more controlled, directed application,

2. the ability to stick seed on a steep slope without the probability of it rolling, blowing, or washing downslope, and

3. the possibility of enhancing germination and growth of the seed through the addition of water and fertilizer during the act of seed application.

Known disadvantages are that it is usually only applied from a truck and thus is appropriate only in areas with roads and it is expensive compared to aerial seeding. Because this technique is relatively new, the effects of hydroseeding on the native vegetation have not been explored in any detail. However, preliminary observations after the Oakland fire (Booker et al. 1993) suggest that the effect on at least initial germination of species in the existing seed bank is inhibitory, and the hydromulch may actually increase the probability of shallow slope failures on fine-grained soils by increasing soil moisture through the reduction of surface runoff. Obviously more research needs to be done on this technique.

Conclusions

When should we seed?

Enough evidence exists to strongly question the utility of seeding from the standpoint of erosion control. The unpredictability of rainfall and its obvious correlation with successful seeding strongly supports a reduction in its use where a reliable erosion control is necessary (Beyers et at. 1994). The effects of competition by Italian ryegrass on native species has also been clearly demonstrated. The value of seeding with any annual grass species for erosion control is in serious question. Because of the dangers of type-conversion land managers should be very cautious about the use of seeding. Erosion rates do increase under many conditions in grassland converted from scrubland. In addition, although successful seeding may reduce surface erosion from rilling and sheeting, it may concurrently increase the rates of erosion due to bioturbation and shallow slope failure, and in the long run may increase erosion from dry ravel and associated debris torrents. Seeding, no matter how successful, is not effective in reducing the principal cause of erosion in southern California, dry ravel and stream channel scouring.

Seeding with non-natives should be avoided. Even Zorro fescue, a short-statured relatively-non-competitive annual can locally dominate and form dense fine fuels that would be subject to flash fires. Other relatively benign non-natives such as the nitrogen-fixing Hykon rose clover could be eliminated in favor of seeding with native species with similar values such as deerweed *(Lotus scoparius)*, or other native nitrogen fixers appropriate to the burn site.

Seeding with natives could be appropriate for sites that were severely degraded with a near total loss of the native seed bank. However, this should only be considered if the mix used is locally collected and reflecting the pre-fire components of the ecosystem. For this kind of appropriate seeding seed stores must be developed that cater to local geographic regions and to known differences in slope, climate, substrate, and other environmental determinants of species composition. Implicit in this type of seeding is the knowledge and attempted emulation of the species composition of the particular stand being treated.

The view of appropriate seeding should also extend to the proper situations. Seeding even on high hazard, erosive sites (as above reservoirs and houses), is problematic due to vagaries of climate for vegetation establishment, and because some types of steep, unstable slopes may not ever be amenable to any form of vegetation augmentation as a useful erosion control device. If the problems are truly localized, as on certain cut banks above road beds or on certain steep unstable slopes where the native seedbank has been removed, seeding at least by aerial application is probably not the appropriate mechanism for erosion control. Mechanical means such as straw netting, plastic sheeting, temporary sediment basins, or permanent slope re-engineering are more reliable and in almost all cases probably more cost- effective approaches for long-term reduction of sediment yield. The slopes in question should be assessed on a case-by-case basis with specific prescriptions written and acted upon individually.

Likewise, seeding with any form of grass should be discouraged unless the grass is a native species selected to augment the seed mix including a majority of appropriate annual and perennial herbs and shrubs.

One of the possible uses of native seed in the future is to restore degraded or type-converted vegetation. With the aid of fire and an appropriate seed mix it may be feasible to "re-convert" annual-dominated vegetation back to native-dominated scrub. For example, a locally collected seed bank could be used after a hot fire in annual grassland to initiate shrub growth by coastal sage scrub pioneer species or seeding could be used to enhance early season (spring or late winter) burns which would otherwise have a high probability of poor regeneration (Parker 1989). However, there are many concerns to be addressed when considering the restoration of a habitat of such spatial and temporal complexity as southern California chaparral or coastal sage scrub (Read and Griswold 1992).

Changes afoot in the agencies

We need to move beyond the scope of seeding as a critical issue in post-fire rehabilitation. There are many other problems of greater concern that need to be dealt with. Some of the most promising research involves modeling the system from hydrological and ecological standpoints. Viewing watersheds from the standpoint of fuel loading, slope hazards, current vegetative cover, and developing a well integrated sense of decision rules about these conditions and appropriate actions will lead to an integrated, logical approach to post-fire management.

There is growing interest and energy among many agencies responsible for various aspects of fire and natural resource management to work together and forge scientifically founded, well-integrated response networks able to make well informed, timely post fire management decisions. Currently the Department of Fish and Game and the California Department of Forestry are working on a joint fire policy that will outline a new approach to pre-, during-, and post-fire vegetation and ecosystem management. The portion of the policy that deals with seeding will reflect the new thinking in these departments, stressing the use of the most recent scientific findings in implementing the fire policy. It will take a conservative approach and will distinguish only certain circumscribed situations where seeding is appropriate.

Prior agency response to emergency watershed protection following fire has often assumed that large percentages of fire areas required treatment (e.g., Ruby 1989). Because of the absolute size of many California brush land fires (see Minnich 1994) cost-effective and expeditious approaches embodying the large-scale rehabilitation philosophy are essentially limited to aerial reseeding with ryegrass or other cheap abundantly available seed. However, we believe that much more effective and appropriate erosion control measures can be taken by focusing on critical problem areas. Thus, we can arrive at very detailed and cost-effective prescriptions for post-fire management since only a small subset of the effected area is likely to be actively managed. A similar type of focused assessment independent of the agency response was applied to the 1991 Oakland fire, with substantial savings of time and money identified (Spittler 1993).

As a part of this policy the Department of Fish and Game is currently developing a conceptual model of how post fire management should proceed,. A simplified flow chart depicting the kinds of decisions made in the model is shown in Figure 1. Some of the suggested actions on this chart are not substantiated by

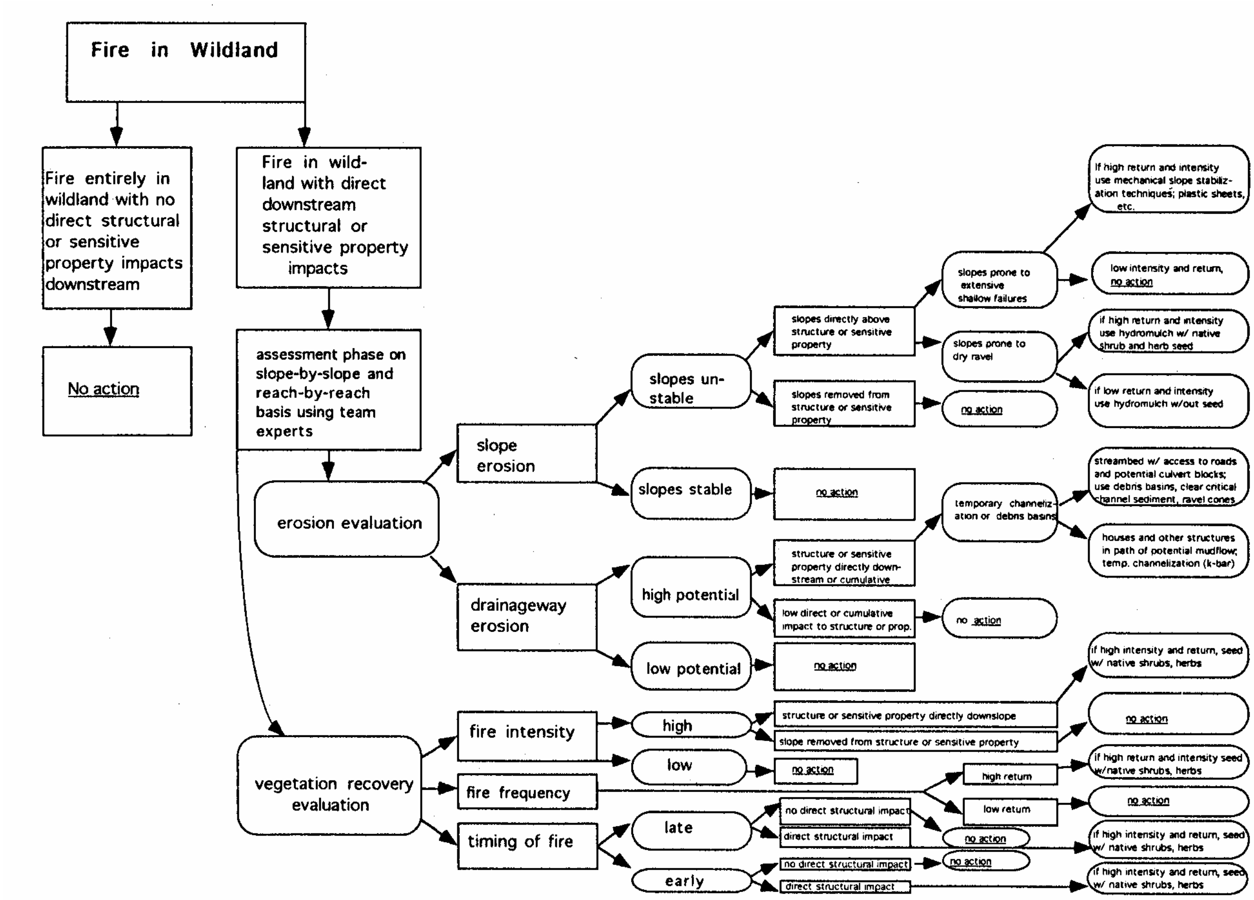

Figure 1. Simplified flow diagram for post-fire management procedures.

available research (for example, hydroseeding) and thus, these approaches will be validated only by future research.

The model makes several assumptions:

1. No active management should be taken on fires that are completely within a wildland area with no direct down-slope or down-stream impacts to sensitive property or structures.

2. The need for post-fire management with some down slope or down stream impacts to property or structures should be assessed by a team of experts including experts in hydrology, geology, plant ecology, and fire ecology.

3. This team should make assessments on a slope-by-slope and stream reach-by-stream reach basis, evaluating each part of the fire zone for erosion hazard and vegetation recovery potential.

4. The evaluations of vegetation and erosion hazard should be made on the basis of partitioning the fire zone into slope- aspect polygons. The team should also evaluate each polygon unit on the basis of the within-slope variations in slope stability and vegetation recovery potential.

5. Drainageway erosion considerations should be judged by cumulative upstream effects as well as up-slope erosion potential. They should also be judged by the direct impacts they will have on sensitive structures and property.

6. Information on rare and sensitive species, natural communities, and other sensitive resources will influence the post-fire management actions.

7. Hazard mitigation should be implemented at the most strategic points within the drainage. These points will be determined by accessibility, as well as effectiveness of placement.

8. The ultimate decision of whether to act or not will involve an interplay of decisions by the team on the interactive effects of slope erosion, drainageway erosion, fire intensity, fire frequency, history of the affected vegetation stand, and seasonal timing of the fire.

9. Decisions are made involving the interactions of both erosion and vegetation recovery concepts. For example decisions to take no action regarding an erosion-related symptom are superseded if there is a vegetation-related symptom which requires some specific action.

10. Decisions that involve seeding with woody plants and herbs should be understood to involve only native species that are collected from sources and environments that are known to be appropriate to the site.

11. The plant ecologist on the evaluation team will need to determine the pre-fire vegetation, the appropriate species to seed with, and the most appropriate lot of seeds to use.

12. Without the benefit of an understanding of the genetic variation involved, seed banks should be collected on a regional basis (e.g., there should be a localized seed bank for each natural region such as the San Gabriel Mountains or coastal San Diego County within the southwest ecoregion).

13. The erosion mitigation will be monitored for effectiveness and maintenance at regular intervals.

14. Erosion hazards will also dictate long-term decisions to alter location of structures (e.g., roads) and future development projects. This will also require the implementation of a detailed hazard assessment program for all urban-wildland interfaces. This program will include a GIS-based modeling of the watershed's hydrology, slope stability, erosion rates, accurate vegetation mapping on a large enough scale to identify dominant species composition on individual slopes, and regular updating of vegetation maps and site history data.

Incorporated within this revised approach should also be some level of rational humility. We must recognize that despite our scientific advances, there is only so much science and agency response can do to mitigate the losses to property and life due to fires. Fires and erosion are natural processes and we cannot and should not try to halt them or diffuse them to the point of massive habitat conversion. Natural rates of erosion have increased dramatically immediately following fires for thousands of years in southern

California. We can no more expect to halt this process than we can expect to halt the periodic tectonic movements that build up the eroding mountains. Instead we need to learn how to live within the natural realm so that natural disturbance will be allowed to perpetuate the cycles of the locally adapted ecosystems.

There are a number of critical social issues that need to be addressed in creative and sensitive ways before we can come to coexist with the natural shrub ecosystems of southern California. Choices of appropriate places to live and appropriate uses of various areas should be made based on the capacity of the landscape and not on our whims and the outmoded concept that humans can manipulate nature as they see fit without any consequences.

Acknowledgments. I am indebted to Jon Keeley for valuable suggestions on how to improve this paper and thank Boyd Gibbons and Susan Cochrane of the Department of Fish and Game for giving me free rein to say what I feel.

Literature Cited

Aiken, S.G. and R.I. Lonard. 1993. *Vulpia,* pp. 1302-1303. *In* J. Hickman (ed), The Jepson manual: higher plants of California. University of California Press, Berkeley, California.

Anderson, H.W., G.B. Coleman, and P.J. Zinke. 1959. Summer slides and winter scour dry-wet erosion in southern California mountains. USDA Forest Service, Pacific Southwest Forest and Range Experiment Station, General Technical Report PSW- 18.

Anonymous. 1970. Emergency revegetation of burned watersheds – annual report. California Division of Forestry, Sacramento.

Anonymous. 1993. Fact sheet: Zorro annual fescue. USDA Soil Conservation Service, FS-41, Sacramento, California.

Barro, S.C. and S.G. Conard. 1987. Use of rye grass seeding as an emergency revegetation measure in chaparral ecosystems. USDA Forest Service, Pacific Southwest Forest and Range Experiment Station, General Technical Report PSW-102.

Barrows, A.G., *S.S.* Tan, and P.J. Irvine. 1993. Damaging landslides related to intense rainstorms of January-February 1993, southern California. California Geology 46:123-131.

Beyers, J.L., S.G Conard, and C.D. Wakeman. 1994. Impacts of an introduced grass seeded for erosion control, on postfire community composition and species diversity in southern California chaparral, pp 594-601. In Proceedings of the 12th Conference of Fire and Forest Meteorology, Society of American Foresters, Bathesda, Maryland.

Blankenbaker, G. and T. Ryan. 1985. Aguanga bum soil erosion and vegetation recovery – administrative study. USDA Forest Service, Cleveland National Forest, unpublished document.

Booker, F.A. W.E. Dietrich, and L.M. Collins. 1993. Runoff and erosion after the Oakland Firestorm. California Geology 46:159-173.

Borchert, M. 1995. Fire intensity and vegetation recovery in chaparral: a review, pp. 91-100. In J.E. Keeley and T. Scott (eds), Brushfires in California wildlands: ecology and resource management. International Association of Wildland Fire, Fairfield, Washington.

Clark, C. 1993. How dry is a drought? Crossosoma 19(2):37 48.

Conrad, C.E. 1979. Emergency post-fire seeding using annual ryegrass, pp. 5-8. In CHAPS Newsletter. USDA Forest Service, Forest Fire Laboratory, Riverside, California.

Corbett, E.S. and L.R. Green. 1965. Emergency revegetation to rehabilitate burned watersheds in southern California. USDA Forest Service, Pacific Southwest Forest and Range Experiment Station, Research Paper PSW-22.

Corbett, E.S. and R. Rice. 1966. Soil slippage increased by brush conversion. USDA Forest Service, Pacific Southwest Forest and Range Experiment Station, Research Notes PSW-128.

Corbett, E.S. and R.P. Crouse. 1968. Rainfall interception by annual grass and chaparral. USDA Forest Service, Pacific Southwest Forest and Range Experiment Station, Research Paper PSW-48.

DeBano, L.F. R.M. Rice, and C.E. Conrad. 1979. Soil heating in chaparral fires: effects on soil particles, plant nutrients, erosion, and runoff. USDA Forest Service, Pacific Southwest Forest and Range Experiment Station, Research Paper PSW-145.

Edmunson, G. and D. Cornelius. 1961. Ryegrass for southern California fuelbreaks. USDA Forest Service, Pacific Southwest Forest and Range Experiment Station, Research Note PSW -186.

Gautier, C.R. 1982. The effects of ryegrass on erosion and natural vegetation recovery after fire, p. 599. In C.E. Conrad and W. Oechel (eds), Dynamics and management of mediterranean-type ecosystems. USDA Forest Service, Pacific Southwest Forest and Range Experiment Station, General Technical Report PSW-58.

Gautier, C.R. and P.H. Zedler. 1982. Does emergency revegetation reduce or increase sediment yields from chaparral watersheds?, pp. 6-8. In Chaparral resource management newsletter Number 1 (Fall 1982). University of California Cooperative Extension, San Diego.

Haidinger, T.L. and J.E. Keeley. 1993. Role of high fire frequency in destruction of mixed chaparral. Madroño 40:141-147.

Heady, H.F. 1977. Valley grasslands, pp. 491-514. In M.G. Barbour and J. Major (eds), Terrestrial vegetation of California. John Wiley & Sons, New York.

Hickman, J. (ed). 1993. The Jepson manual: higher plants of California. University of California Press, Berkeley.

Horton J.S. and C.J. Kraebel. 1955. Development of vegetation after fire in the chamise chaparral of southern California. Ecology 36:249-262.

Jepson, W.L. 1925. Manual of the flowering plants of California. University of California Press, Berkeley.

Keeley, J.E. 1990. The California valley grassland, pp. 3-23. In A.A. Schoenherr (ed), Endangered plant communities of southern California. Southern California Botanists, Fullerton, Special Publication Number 3.

Keeley, J.E. 1991. Seed germination and life history syndromes in California chaparral. Botanical Review 57 :81-116.

Keeley, J.E., M. Carrington, and S. Trnka. 1995. Overview of management issues raised by the 1993 wildfires in southern California. Pp 83-89. In J.E. Keeley and T. Scott (eds), Brushfires in California wildlands: ecology and resource management. International Association of Wildland Fire, Fairfield. Washington.

Keeley, J.E. and S.C. Keeley. 1984. Postfire recovery of California coastal sage scrub. American Midland Naturalist 111:105-117.

Keeley, J.E. and S.C. Keeley. 1988.Chaparral,pp.165-208.In M. G. Barbour and W.D. Billings (eds), North American terrestrial vegetation. Cambridge University Press, Cambridge, U.K.

Keeley, J. and P. Zedler. 1978. Reproduction of chaparral shrubs after fire: a comparison of sprouting and seeding strategies. American Midland Naturalist 99:142-161.

Keeley, S.C., J.E. Keeley, S.M. Hutchinson, and A.W. Johnson. 1981. Postfire succession of the herbaceous flora in southern California chaparral. Ecology 62: 1608-1621.

Krammes, J.S. and L.W. Hill. 1963. First-aid for burned watersheds. USDA Forest Service, Pacific Southwest Forest and Range Experiment Station, Research Note PSW-29.

Libby, W.J. and K.A. Rodrigues. 1992. Revegetating the Oakland- Berkeley hills burn. Fremontia 20(1):43-49.

Malanson, P.W. and J. O'Leary. 1982. Post-fire regeneration strategies of Californian coastal sage shrubs. Oecologia 59:355-358.

Matthews, M.A. 1986. The impacts of ryegrass seeding on plant communities of the central coast: a literature review. Graduate studies report, California Polytechnic University, San Luis Obispo.

Minnich, R.A. 1995. Fuel driven fire regimes of the California chaparral, pp. 21-27. In J.E. Keeley and T. Scott (eds), Brushfires in California wildlands: ecology and resource management. International Association of Wildland Fire, Fairfield, Washington.

Munz, P .A. 1968. A California flora and supplement. University of California Press, Berkeley.

Munz, P.A. 1974. A flora of southern California. University of California Press, Berkeley.

NCCP. 1993. Southern California coastal sage scrub natural communities conservation plan. Scientific review panel conservation guidelines and documentation, California Department of Fish and Game, Sacramento.

Naqvi, H.H. 1969. Allelopathic effects of *Lolium multiflorum*. Ph.D. dissertation, University of California, Santa Barbara.

Nelson, L.R. 1980. Recurrent selection for improved rate of germination in annual ryegrass *(Lolium multiflorum).* Crop Science 20:219-221.

O'Leary, J.F. 1990. Californian coastal sage scrub: general characteristics and considerations for biological conservation, pp. 24-41. In A.A. Schoenherr (ed), Endangered plant communities of southern California. Southern California Botanists, Fullerton, Special Publication Number 3.

O'Leary, J.F. 1995. Potential impacts of emergency seeding on cover and diversity patterns of Californian shrubland communities, pp 141-148. In J.E. Keeley and T. Scott (eds), Brushfires in California wildlands: ecology and resource management. International Association of Wildland Fire, Fairfield, Washington.

Odion, D.C. and N.M. Nadkarni. 1985. The effects of seeding an exotic grass *(Lolium multiflorum)* on native seedling regeneration following fire in a chaparral community, pp. 115- 121. In J.J. De Vries (ed), Proceedings of the chaparral ecosystems research conference. University of California, Davis, Water Resources Center, Report 62.

Ordano, J. 1987. Wildfire aftermath: gains and losses. Out door California 48 (6):5-8

Papanastasis, V. 1973. Decline of annual ryegrass *(Lolium multiflorum* Lam.) seeded on burned brushlands. Ph.D. dissertation, University of California, Berkeley.

Papanastasis, V. and H.H. Biswell. 1975. Seeding rate-cover relationships of annual ryegrass seeded on burned brush lands. Journal of Range Management 28:402-406.

Parker, V. T. 1989. Maximizing vegetation response on management bums by identifying fire regimes, pp. 87-92. In N. Berg (ed), Proceedings of the symposium on fire and watershed management. USDA Forest Service, Pacific Southwest Forest and Range Experiment Station, General Technical Report PSW 109.

Patric, J.H. and T.L. Hanes. 1964. Chaparral succession in a San Gabriel Mountains area of California. Ecology 45:353-360.

Read, E. and M. Griswold. 1992. Restoration of coastal sage scrub in southern California – what are we restoring (an open letter)? Society for Ecological Restoration, California Chapter Newsletter Spring 1992.

Rice, R.M. 1974. The hydrology of chaparral watersheds, pp 27-34. In M. Rosenthal (ed), Symposium on living with the chaparral-proceedings. Sierra Club, San Francisco.

Rice, R.M and G.T. Foggin III. 1971. Effect of high intensity storms on soil slippage on mountainous watersheds in southern California. Water Resources Research 7:1485-1496.

Ruby, E.C. 1989. Rationale for seeding grass on the Stanislaus complex bum, pp. 125-130. In N.H. Berg (ed), Proceedings of the symposium on fire and watershed management. USDA Forest Service, Pacific Southwest Forest and Range Experiment Station, General Technical Report PSW 109.

Rundel, P.W. and D.J. Parsons. 1983. Post-fire uptake of nutrients by diverse ephemeral herbs in chamise chaparral. Oegologica 61:285-288.

Sampson, A.W. 1944. Plant succession and burned chaparral lands in northern California. University of California, Berkeley Agricultural Experiment Station, Bulletin 685.

Sawyer, J. 1994. A manual of California vegetation: series level descriptions, first edition. Unpublished manuscript.

Schultz, A. W. and H.H. Biswell. 1952. Comparison between grasses reseeded on burned brushlands in California. Journal of Range Management 5:338-345.

Schultz, A.W., J.L. Launchbaugh, and H.H. Biswell. 1955. Relationship between grass density and brush seedling survival. Ecology 36:226-238.

Sharp, R.P. 1978. Coastal southern California. K/H Geology field guide series. Kendall/Hunt, Dubuque, Iowa.

Skinner, M. W. and B.M. Pavlik. 1994. California Native Plant Society's inventory of rare and endangered vascular plants of California. (Fifth edition). California Native Plant Society, Sacramento. Special Publication Number 1.

Spittler, T.E. 1993. Emergency landslide hazard evaluation following the Tunnel Fire October 19-23,1991. California Geology 46:174-179.

Spittler, T .E. 1995. Fire and the debris flow potential of winter storms. pp. 113-120. In J.E. Keeley and T. Scott (eds), Brushfires in California wildlands: ecology and resource management. International Association of Wildland Fire, Fairfield. Washington.

Sweeney, J.R. 1956. Responsesof vegetation to fire: a study of the herbaceous vegetation following chaparral fires. University of California Publications in Botany 28: 143-250.

Taskey, R.D., C.L. Curtis, and J. Stone. 1989. Wildfire, ryegrass seeding, and watershed rehabilitation, pp. 115-124. In N.H. Berg (ed), Proceedings of the symposium on fire and watershed management. USDA Forest Service, Pacific Southwest Forest and Range Experiment Station, General Technical Report PSW 109.

USDA Forest Service. 1994. Policy and guidelines for the uses of native species in habitat restoration and rehabilitation in Region Five. Unpublished document, Regional Office, San Francisco.

Wakimoto, R.H. 1979. Untitled. CHAPS Newsletter. USDA Forest Service Forest, Fire Laboratory, Riverside, California.

Wells, P. V. 1962. Vegetation in relation to geological substratum and fire in the San Luis Obispo quadrangle, California. Ecological Monographs 32:79-103.

Wells, W.G. 1981. Some effects of brushfire on erosion processes in coastal southern California, pp 123-132. In T.R.H. Davies and A.J. Pearse (eds), Erosion and sediment transport in Pacific Rim steeplands symposium. International Association of Hydrological Sciences, Christchurch, New Zealand, Publication Number 132.

Wells, W.G. 1984. Fire dominates sediment production in California chaparral, pp 163-164. In D. Bell (ed), MEDECOS IV, Proceedings of the fourth international conference on Mediterranean ecosystems. University of Western Australia, Netlands.

Westman, W.E. and J.F. O'Leary. 1986. Measures of resilience: the response of coastal sage scrub to fire. Vegetatio 65:179- 189.

Westman, W.E. 1982. Coastal sage scrub succession, pp. 91-99. In C .E. Conrad and W. Oechel (eds), Dynamics and management of Mediterranean-type ecosystems. USDA Forest Service, Pacific Southwest Forest and Range Experiment Station, General Technical Report PSW-58.

Wright, D.L. 1977. Erosion control with vegetation in highway corridors. Ph.D. dissertation, Virginia Polytechnic Institution and State University, Blacksburg.

Zedler, P .H., C.R. Gautier, and G.S. McMaster. 1983. Vegetation change in response to extreme events: the effect of a short interval between fires in California chaparral and coastal scrub. Ecology 69:809-818.

Zedler, P .H. 1995. Fire frequency and vegetation recovery, pp. 101-112. In J.E. Keeley and T. Scott (eds), Brushfires in California wildlands: ecology and resource management. International Association of Wildland Fire, Fairfield, Washington.